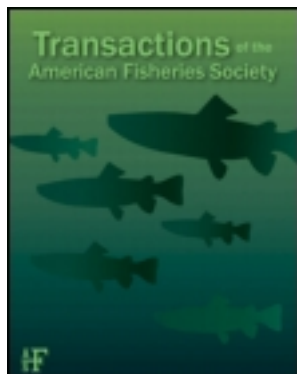


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ARTICLE

Effects of Stocking Catchable-Sized Hatchery Rainbow Trout on Wild Rainbow Trout Abundance, Survival, Growth, and Recruitment

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Abstract

The Idaho Department of Fish and Game has proactively dealt with the potential adverse genetic effects of stocking catchable-sized hatchery trout in waters that support native salmonids by adopting a policy in 2001 whereby only sterile rainbow trout *Oncorhynchus mykiss* are stocked in flowing waters; however, concerns regarding the competitive effects of introducing hatchery trout into streams and rivers supporting wild trout have not been addressed. We stocked fish in the middle 3 years of a 5-year study to assess whether stocking hatchery rainbow trout of catchable size (hereafter, catchables) reduced the abundance, survival, growth, or recruitment of wild rainbow trout in streams. Catchables averaging 249 mm total length (TL) were stocked from 2006 to 2008 at an annual density of 4.2 fish/100 m² into 12 treatment reaches of stream that were paired with control reaches at least 3 km apart in the same stream in which no stocking occurred. Wild rainbow trout abundance (including all fish ≥ 75 mm TL), recruitment, survival, and growth were determined from population estimates and recaptures of fish tagged with passive integrated transponder (PIT) tags during mark–recapture electrofishing sampling. The abundance of wild rainbow trout averaged 13.2 fish/100 m² but varied substantially across sites and years, ranging from a low of 0.5 to a high of 131.3 fish/100 m²; similar variability was observed in recruitment to age 1. Estimates of total annual survival averaged 0.53 based on the population abundance estimates (which allowed for emigration and immigration) and 0.26 based on the PIT-tag recaptures (which allowed for emigration but not immigration). Our paired study design demonstrated that the abundance, survival, growth, and recruitment to age 1 of wild rainbow trout were all unaffected by stocking catchables. The lack of population-level effects from stocking catchables was not surprising considering the high short-term mortality and the socially and physiologically naive behavior typically exhibited by hatchery catchables stocked in lotic systems.

Maintaining put-and-take fisheries in streams, ponds, lakes, and reservoirs that cannot withstand the harvest demands of anglers is a common use of hatchery-reared, catchable-sized fish (Utter 1994; Epifanio and Nickum 1997). In 2008, a total of 2,466,000 catchable-sized fish (catchables) were stocked by the Idaho Department of Fish and Game (IDFG) in waters of Idaho (T. Frew, IDFG, personal communication). Even though 64% of the 323 waters stocked are lentic systems, IDFG stocks more than 500,000 catchables into streams annually. Although much of this stream stocking occurs in reaches where pure native salmonids no longer exist and where environmental conditions can no longer support robust wild salmonid populations, roughly

40% of stream catchables in Idaho continue to be stocked in waters containing native or naturalized nonnative trout (Dillon et al. 2000). Such stocking reflects one end of the dichotomous responsibility of many resource managers in fish and wildlife agencies, that of providing harvestable surpluses to the public while at the same time protecting native species. In some cases, stocking hatchery trout can play a direct role in native or wild trout management by directing consumptive angling to specific streams or stream reaches that provide little to no production of native or wild trout, and this can foster public acceptance of more restrictive regulations on other waters where native trout occur (Van Vooren 1995). Supplementing wild trout stream

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fisheries with hatchery fish, however, has long raised concern over potential adverse genetic and ecological (i.e., competition) effects (Allendorf 1991; Krueger and May 1991; Weber and Fausch 2003).

Regarding genetic effects, the stocking of hatchery trout over the last century has resulted in widespread interspecific and intraspecific hybridization between native and hatchery salmonids in Idaho (e.g., Meyer et al. 2006; Kozfkay et al. 2011) and elsewhere (Young 1995; Allendorf et al. 2001; Simmons et al. 2009). In recent decades, recognition of this problem and concern with the continued spread of hybridization has led to a reduction in trout stocking in some areas. For example, stream stocking of catchable-sized rainbow trout *Oncorhynchus mykiss* (hereafter, catchables) by IDFG from 1985 to 2008 was reduced by 50% in quantity and more than 50% in river kilometers stocked, and now occurs in less than 2% of the 44,000 fishable kilometers in Idaho (IDFG, unpublished data). Moreover, since 2001 IDFG has only been stocking catchables that have been treated to induce sterility (Kozfkay et al. 2006); currently these fish are produced largely from all-female triploid eggs purchased from Troutlodge (Sumner, Washington), where triploid induction rates in recent years has been 100% on all batches tested (A. Barfoot, Troutlodge, personal communication). These steps have probably stemmed the future threat of hybridization from ongoing IDFG stocking practices for catchables, although the further spread of hybridization via dispersal and invasion is likely to continue (e.g., Rubidge and Taylor 2005).

In contrast to these genetic concerns, possible adverse competitive interactions in streams and rivers between wild trout and stocked catchables have not been addressed. Competition, by definition, causes a reduction in fitness of an organism owing to the limited supply of a resource held in common with other organisms, or the limited ability to exploit a resource because of interference by other organisms (Birch 1957). Reduced fitness levels in wild trout populations could translate to decreased survival, growth, or reproduction (Moyle and Cech 1982). Most competition studies on wild trout have indirectly assessed changes in fitness levels, or have found evidence of competition by inferring causal relationships between fitness and characteristics such as the ability to maintain favorable positions (Griffith 1972; Fausch and White 1986; Peery and Bjornn 1996), win agonistic bouts (Griffith 1972; Mesa 1991; McMichael et al. 1999), gain weight (Dewald and Wilzbach 1992; Harvey and Nakamoto 1996), or survive (Kocik and Taylor 1994). These studies at the microhabitat scale are easy to replicate with different manipulations of fish compositions and densities to test interspecific and intraspecific competition. However, they do not directly address concerns at the population level (Fausch 1998), a scale at which competition investigations are rarely performed (Schoener 1983). Of the population-level studies conducted on competition between hatchery and wild trout, the two foremost, of which we are aware, have contradicting conclusions. Vincent (1987) concluded hatchery trout decreased the abundance and biomass of wild rainbow trout and brown trout *Salmo trutta* in

the Madison River and O'Dell Creek, Montana, while Petrosky and Bjornn (1988) concluded that catchables had little effect on wild cutthroat trout *O. clarkii* in the St. Joe River and wild rainbow trout in Big Springs Creek, Idaho.

In IDFG's Fisheries Management Plan, the Department emphasizes using stream catchables for stocking in areas where there is convenient access for anglers, return to creel is good (preferably $\geq 40\%$), and stocking does not negatively affect native species. In those streams in Idaho that support wild salmonids and continue to be stocked with sterile catchables, we questioned whether stocking these hatchery fish would affect wild trout populations at a measurable level and in a meaningful way. Specifically, we tested for population-level competition effects of stocked catchables on wild rainbow trout populations by quantifying the abundance, survival, growth, and recruitment of wild trout populations where catchables were stocked for several years.

STUDY AREA

Eleven study streams in the upper Snake River basin of southern Idaho were used to establish 12 paired treatment and control reaches that ranged from 1,094–2,104 m in elevation, from 0.5% to 6.2% in gradient, and from 30 to 360 $\mu\text{S}/\text{cm}$ in specific conductivity (Table 1; Figure 1). Study reaches were, on average, 719 m in length, which we believed would be long enough to estimate the trout population metrics in question and apply them to the population level. Streams were grouped by three angling regulation categories: general (six fish bag limit), wild trout (two fish limit), and catch and release (Table 1). The three streams in the latter category were not explicitly managed with catch-and-release regulations, but they functioned as such because slow fish growth and a 356-mm minimum length limit meant that nearly all fish were of sublegal size for two streams, and very limited public access in a rugged roadless canyon (surrounded by private property) resulted in essentially no fishing pressure for a third stream. During our analyses, differences in the effects of competition among the three fish-regulation categories were not apparent; hence, results from all study streams were eventually combined. The fishing season for all streams lasted from Memorial Day (last Monday in May) through November 30.

In addition to the angling regulation categories, further selection criteria for study streams were included to simplify the study design. One criterion was that the stream was not already being stocked nearby (i.e., within 5 km of any study reach) and had not been stocked in the last 10 years. Another criterion was that 3 km of stream length could be established between two study reaches (one treatment and one control) on each stream; this distance was assumed to be long enough so that few stocked catchables would move from a treatment to a control site (High and Meyer 2009). A final criterion was for rainbow trout to dominate the salmonid composition of the stream. In our study streams, redband (rainbow) trout *O. mykiss gairdneri* are native

TABLE 1. Study reach locations, biotic and abiotic characteristics, and treatment stocking rates (per month; stocking occurred in the middle of the treatment reaches) for study streams in southern Idaho. Abbreviations are as follows: RBT = rainbow-redband trout, BUT = bull trout, and YCT = Yellowstone cutthroat trout.

Stream	Reach	UTM coordinates			Reach length (m)	Mean width (m)	Elevation (m)	Drainage area (km ²)	Conductivity (μS/cm)	Mean summer water temperature (°C)	Stream order (1:100,000)	Gradient (%)	Percent substrate composition			Native trout	Catchables/plant
		Eastings	Northings	Zone									Fines	Gravel	Cobble/boulder		
General regulation streams																	
1. Fourth Fork Rock Creek	Treatment	726775	4681619	11	812	3.6	1,577	25	65	13.6	3rd	4.8	4	34	53	RBT	51
	Control	725117	4678773	11	443	3.0	1,800	14	58	12.9	2nd	3.5	5	39	44		
2. East Fork Weiser River	Control	550180	4961897	11	489	5.5	1,260	70	86	13.1	2nd	3.3	6	20	53	RBT/	39
	Treatment	553858	4962111	11	495	6.5	1,481	50	62	9.8	2nd	5.9	5	23	63	BUT	
3. Little Weiser River	Treatment	555728	4927393	11	641	10.6	1,163	116	81	15.7	3rd	1.8	5	27	60	RBT/	70
	Control	560119	4929915	11	581	8.7	1,306	85	70	12.4	3rd	2.2	2	9	73	BUT	
Wild trout regulation streams																	
4. Second Fork Squaw Creek	Control	555742	4913591	11	873	8.7	1,149	96	39	15.8	3rd	1.4	4	37	49	RBT/	88
	Treatment	556947	4920060	11	684	9.9	1,254	66	38	14.9	3rd	1.8	2	29	57	BUT	
5. Clear Creek	Treatment	612447	4884548	11	545	11.2	1,267	136	44	12.2	3rd	3.1	6	18	63	RBT/	124
	Control	615483	4892526	11	371	10.5	1,562	93	33	10.4	3rd	2.5	4	12	62	BUT	
6. Sawmill Creek	Treatment	313446	4909346	12	707	8.2	2,036	192	69	12.4	4th	1.3	7	35	48	BUT?	57
	Control	310744	4914244	12	521	6.4	2,104	116	53	11.1	4th	1.4	8	35	46		
7. Willow Creek	Treatment	691836	4817640	11	461	4.7	1,717	51	244	12.0	3rd	2.5	28	41	15	RBT	23
	Control	690451	4819520	11	349	3.6	1,790	31	255	11.1	3rd	2.9	7	42	39		
8. Medicine Lodge Creek	Treatment	380935	4904871	12	567	5.2	1,737	409	392	13.0	4th	1.6	26	30	7	YCT	44
	Control	376226	4907857	12	527	5.9	1,806	394	380	12.5	4th	1.7	18	29	19		
Catch-and-release regulation streams																	
9. South Fork Boise River	Control	660998	4827702	11	1,439	23.3	1,343	917	100	13.4	5th	1.6	3	25	63	RBT/	227
	Treatment	664707	4828430	11	1,084	23.2	1,604	899	106	15.3	5th	1.7	7	35	43	BUT	
10. Middle Fork Boise River	Treatment	613554	4843073	11	827	26.9	1,094	984	68	18.1	5th	0.5	2	27	47	RBT/	303
	Control	618340	4848149	11	915	30.8	1,171	905	75	15.6	5th	0.5	8	19	57	BUT	
11. Middle Fork Boise River	Treatment	626494	4849524	11	964	23.0	1,269	757	64	15.1	5th	0.4	3	16	59	RBT/	270
	Control	631618	4852102	11	925	24.7	1,305	647	60	14.5	5th	1.0	4	16	61	BUT	
12. Badger Creek	Control	477674	4862215	12	217	11.9	1,646	150	248		3rd	4.4	3	19	46	YCT	
	Treatment	480530	4863772	12	470	16.0	1,698	145	224	10.8	3rd	0.9	6	39	43		181

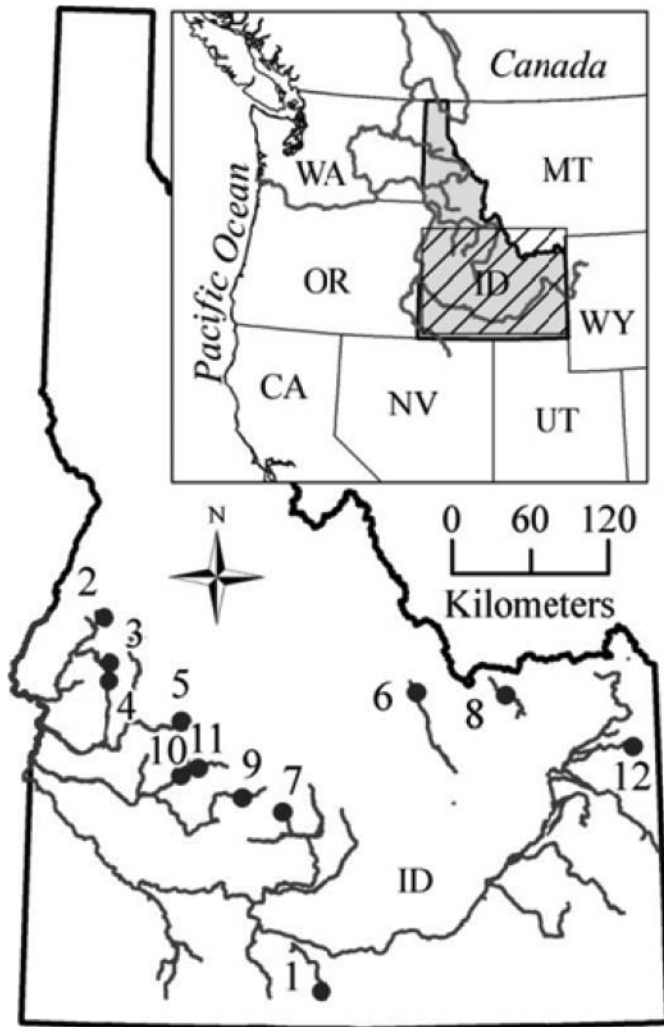


FIGURE 1. Locations of study streams in southern Idaho. Study site numbers correspond to those on the far left column of Table 1.

to 8 of the 11 study streams and historically were probably the dominant trout species present in these streams (Table 1). Kozfkay et al. (2011) found that native rainbow–redband trout had hybridized with nonnative rainbow trout (of coastal origin, via historical stocking practices) in two of our test streams (Second Fork Squaw and Willow creeks), were pure in three streams (East Fork Weiser River, Little Weiser River, and Clear Creek), and were untested in the remaining three streams where they were native. Bull trout *Salvelinus confluentus* are native to six or seven streams and are generally found sympatrically with rainbow trout, although currently (and probably historically) at much lower densities. In the remaining two streams, Yellowstone cutthroat trout *O. clarkii bouvieri* are native, although rainbow trout have largely or entirely replaced them in our study reaches. During our study, rainbow trout, on average, comprised 94% of the trout population in our study reaches, followed by cutthroat trout (3%), brook trout *S. fontinalis* (2%), bull trout (1%), and brown trout (<1%). Other species we en-

countered included mountain whitefish *Prosopium williamsoni* and several species of Cottidae, Catostomidae, and Cyprinidae.

METHODS

From 2006–2008, catchable rainbow trout were stocked into the middle of treatment reaches three times during the growing season at monthly intervals at a mean monthly stocking density of 1.4 fish/100 m². The resulting annual stocking density (4.2 fish/100 m²) and the monthly stocking intervals were based on typical stream stocking by IDFG; density was selected to be at the high end of ongoing stocking rates to serve as a worst-case scenario for potentially negative competition effects on wild trout populations. Moreover, an equal number of catchables to those stocked in the middle of the stocked reaches were also planted at the upstream- and downstream-reach boundaries to account for dispersal in order to maintain an elevated catchable density. In 2006, mean \pm SD size (total length [TL]) of a subsample of stocked catchables was 249 \pm 31 mm (range, 89–377 mm; n = 853), and, based on hatchery records, year-to-year variation in fish size at stocking was minimal. To avoid affecting wild trout populations solely by increasing harvest after catchables were stocked (Butler and Borgeson 1965; Carline et al. 1991; Baer et al. 2007), we did not advertise any stocking events or locations. The treatment (i.e., stocking) reach in each stream was randomly assigned to one of the paired reaches except at Badger Creek, where logistical constraints of planting catchables required that the upper site serve as the treatment.

From 2005–2009, which incorporated 1 year before and after stocking, salmonid populations were sampled with backpack and canoe-mounted electrofishing gear in order to conduct mark–recapture population sampling. To minimize the effect that seasonal changes can have on fish abundance (Decker and Erman 1992), paired sites (i.e., controls and treatments) were sampled in the same week, and sampling was repeated each year within a few weeks of the same calendar date for each paired location. All captured salmonids were identified to species, measured to the nearest millimeter (TL), and weighed to the nearest gram by using a top-loading digital scale. Fish scales were collected from a subsample of fish for aging purposes. Passive integrated transponder (PIT) tags (12 mm long, 2 mm in diameter; Destron Fearing) were implanted intraperitoneally in most captured wild rainbow trout 75 mm and larger from 2006 to 2008 to estimate growth and survival in subsequent years. For each reach, one recapture run was made 1–2 d after the marking run.

We used the Fisheries Analysis Plus program (Montana Fish, Wildlife, and Parks 2004) to calculate population estimates and 95% confidence intervals (CIs) with the Lincoln–Petersen mark–recapture model as modified by Chapman (1951). In nearly every instance, we were able to create size-classes (generally 25–50 mm) meeting the criteria that (1) the number of fish marked in the marking run multiplied by the catch in the recapture run was at least four times the estimated population

size, and (2) at least three recaptures occurred per size-class. Meeting these criteria creates modified Petersen estimates that are biased by less than 2% (Robson and Regier 1964). Estimates were run separately for each species when possible. Size of age-0 rainbow trout varied considerably between streams over the course of each year and between sample years, but generally they were less than 75 mm TL. For consistency, 75 mm was subsequently used as the cutoff length for population estimates to minimize bias from the variability in size of age-0 fish.

As an additional index of population size, we calculated percent habitat saturation (PHS) as developed by Grant and Kramer (1990):

$$\text{PHS} = \sum_{i=1}^n D_i \times T_i \times 100 \times 1.19,$$

where D_i is the density of fish (fish/m²) of size-class i and T_i is the territory size of size-class i , and 1.19 is a correction factor to remove the bias introduced from the log transformation in territory size. Territory size was estimated from Grant and Kramer (1990) as

$$\log_{10}(T_i) = 2.61 \times \log_{10}[\text{fork length (cm)}] - 2.83,$$

where fork length (FL) was the average FL for size-class i . A PHS of 100 means that salmonids fill the surface area of the stream bottom, assuming that territory size reasonably represents the salmonid space needs in the stream. We included PHS because it incorporates several parameters (i.e., fish size, fish density, and territoriality) that probably reflect carrying capacity within study reaches better than density alone (Grant and Kramer 1990). Because we recorded data as TL, FL for rainbow trout was calculated with the equation $\text{FL} = \text{TL}/1.049$ (Carlander 1969).

Scale aging was used to separate age-1 fish from other cohorts. Otoliths were not used because killing fish each year to obtain otoliths could have altered population densities and other population metrics in subsequent years. Although scales usually underestimate age for older rainbow trout (e.g., Hining et al. 2000; Cooper 2003), they tend to be a reasonably accurate aging structure in southern Idaho for rainbow trout age 2 or less (Schill et al. 2010). Moreover, we assumed that any directional bias in scale aging would not have differed between treatment and control reaches. Scales were independently aged by two readers with no knowledge of fish length. Disparities were reconciled by a third reader who used all available information (including fish length for the scale being aged, fish length frequency at the site, and sampling date) to agree on a final age. We compared aging agreement between the two initial readers by calculating the between-reader coefficient of variation ($\text{CV} = \text{SD}/\text{mean} \times 100$) (Chang 1982) for each site in each year, which ranged from 0.6% to 45.1% and averaged 10.7%. Because length

usually overlapped slightly from age 0 to age 1 and from age 1 to age 2, age-length keys with 10-mm length-groups were used to allocate ages (i.e., age 0, age 1, or age 2 and older) for all fish in the overlapping length-groups to develop size cutoffs before population estimates were developed.

Using this scale-aging information, we computed separate population estimates for age-1 and older fish and age-2 and older fish. This allowed us to estimate total annual survival (S), which was calculated for each year with Heinicke's estimator $S = (N - N_1)/N$ (Ricker 1975), where N and N_1 are the abundance of age-1 and older fish and age-2 and older fish, respectively. Survival for fish age 1 and older was assumed to be constant across all age-classes. Following Carline (2006), we used the variance from the population estimates to estimate the variance of S by employing a Taylor series approximation (Som 1996). Recruitment to age 1 was estimated by computing population estimates for fish only within the age-1 size-class.

For a second estimate of survival, we used capture histories of PIT-tagged fish to estimate apparent survival (ϕ) and recapture (p) probabilities by using open population Cormack-Jolly-Seber (CJS) models (Lebreton et al. 1992) in Program MARK (White and Burnham 1999). We used an information-theoretic approach based on Akaike's information criterion corrected for small sample size (AIC_c) (Burnham and Anderson 2002) to rank candidate models. We constructed single stage models (i.e., fish of all stages grouped together) that had stream reach and treatment effects. We determined goodness of fit of the global model, $\phi(a \times g \times t)$, $p(a \times g \times t)$, by estimating the overdispersion parameter (median \hat{c}) by means of a simulation procedure in Program MARK, where a = area (stream reach), g = treatment or control group, and t = time (year). We also constructed a more parsimonious additive model $\phi(a + g + t)$, $p(a + g + t)$ plus 10 additional models that were more parsimonious than the global model. Since we did not PIT-tag fish in 2005, and ϕ is not estimable between the last two encounters because it is confounded with p in the CJS model (Amstrup et al. 2005), only two estimates of ϕ could be made for each study reach (2006–2007 and 2007–2008). There was no evidence of overdispersion in the PIT tag recapture data, as evidenced by the global model median \hat{c} value of 1.07; we therefore did not adjust estimated standard errors nor model-selection parameter estimates. Only results from the global model were used to estimate ϕ since it had 94% of the model weight (second best model was 5.62 ΔAIC_c units lower).

Our methodology assumed that within-stream rates of recruitment, survival, emigration, and immigration were consistent for each of the control and treatment reach pairs among years, and that any differences between control and treatment pairs in the magnitude of fluctuations in abundance, recruitment, and survival were due to stocking. For estimates of ϕ , we assumed no PIT tags were lost or failed, and that tagging resulted in no mortality. Although we did estimate annual PIT tag loss from 2006 to 2007, we could not estimate annual loss in later years because, after 2007, fish with missing tags could not be

connected to a specific year. Thus, we did not correct ϕ for PIT tag loss.

Data analysis.—All statistical analyses were performed with SAS statistical software (SAS Institute 1999) at $\alpha = 0.05$. To assess the effects of stocking on abundance, PHS, recruitment, and survival, we used repeated measures analysis of variance (ANOVA). The experimental unit was each survey that we conducted within each study reach, and we used year and stream as blocking factors to control for the variation in the response variable owing to these effects. A statistically significant interaction between stocking treatment (stocked or unstocked) and year was used to indicate a stocking effect on the response variable. Because the CJS model only generated two estimates of ϕ (2006–2007 and 2007–2008), we used a paired t -test rather than repeated measures to assess differences in ϕ between treatment and control groups.

To assess possible effects of stocking on growth, we used analysis of covariance (ANCOVA) on PIT tag recapture data. The experimental unit was each recaptured PIT-tagged fish ($n = 1,612$) and the response variable was the growth (in millimeters) from year x to year $x + 1$. We used year and stream as blocking factors and initial fish length (at tagging) as a covariate, and tested for differences in growth between treatment and control reaches. We also calculated relative weight (W_r) according to Anderson and Neumann (1996) and assessed differences in fish condition by comparing mean W_r between treatments and controls with a paired t -test.

As an additional assessment of fish growth, we compared mean length at age between control and treatment reaches. All PIT-tagged fish were retained in the recapture run in 2009 ($n = 385$); they were euthanized by exposure to an overdose of tricaine methanesulfonate (MS-222) in solution and transported to the laboratory where otoliths were removed. Additional fish at some locations were also retained for aging because sample size was inadequate at a number of reaches where only PIT-tagged fish were used. Age estimates were derived by examining digital images of whole otoliths taken with a Leica DC 500 digital camera mounted on a Leica DM4000B compound microscope at $40\times$ magnification. Otoliths were immersed in water and illuminated under oblique, reflected fiber optic light. The outer edge of each translucent zone was counted as an annulus and fish were assumed to reach age 1 on 1 January (DeVries and Frie 1996). As with scales, otoliths were independently aged by two readers who had no knowledge of fish length, and disparities were reconciled by a third reader who used all available information (including fish length, and fish length frequency at the site) to agree on a final age. Aging agreement between the two initial readers was assessed by calculating for each site the between-reader CV, which ranged from 0.0% to 22.7% and averaged 9.6%. Mean length at age 2 and age 3 were calculated for all reaches where at least three fish from each age were available. Mean length at age was compared between control and treatment study reaches with a paired t -test. We only used age 2 and age 3 for these analyses because (1) these fish had

been alive for two or three summers of stocking, and (2) there tended to be adequate sample sizes for these ages compared with older fish.

RESULTS

Density of all wild rainbow trout among study streams averaged 13.2 fish/100 m² and ranged from 0.5 to 131.3 fish/100 m² throughout the study (Figure 2). Trout density was generally highest in Rock Creek (mean = 41.1 fish/100 m²) and Willow Creek (mean = 24.4 fish/100 m²) and lowest in the Middle Fork Boise River (mean = 1.6 fish/100 m²). Over the study period, mean trout density increased on average from 11.4 fish/100 m² in 2005 to 19.5 fish/100 m² by 2007, then decreased to 11.3 fish/100 m² in 2009. During the three stocking years, treatment reaches were on average 22% higher in abundance of wild rainbow trout than in the unstocked years, compared with an average of 24% higher for control reaches. Accordingly, the density of wild rainbow trout was unaffected by the stocking of hatchery catchables (repeated measures ANOVA: $F = 1.23$, $df = 23$, $P = 0.37$). Mean PHS among all sites was 19.8%; PHS ranged from 1.4% to 80.9% (Figure 3) and was also unaffected by stocking catchables (repeated measures ANOVA: $F = 1.60$, $df = 23$, $P = 0.27$).

Recruitment to age 1 increased initially from an average of 6.4 fish/100 m² in 2005 to 12.4 fish/100 m² by 2007, then declined to 5.4 fish/100 m² by 2009 (Figure 4). Most of this increase in 2007 was due to the large increase at the control reach in Willow Creek, partly because beaver *Castor canadensis* activity in this reach resulted in altered stream width in some years; without this site, mean recruitment to age 1 in 2007 was 7.2 fish/100 m². Recruitment to age 1 was relatively consistent from year to year and the CV for recruitment at each site averaged 48.8%, but recruitment was not as consistent as total abundance (average CV = 36.8%). Recruitment of wild rainbow trout to age 1 was not affected by the stocking of hatchery catchables (repeated measures ANOVA: $F = 0.29$, $df = 23$, $P = 0.88$).

Estimates of survival (S) based on scale aging and population abundance estimates ranged widely, from 0.00 to 0.99, and with a mean across all study reaches and years of 0.55 (Figure 5). Mean S from 2005 to 2009 was 0.63, 0.49, 0.53, 0.57, and 0.48 for the 5 years, respectively, for the control group (overall mean = 0.54), compared with 0.61, 0.54, 0.51, 0.62, and 0.51 for the treatment group (overall mean = 0.56). Stocking hatchery catchables did not have any effect on S (repeated measures ANOVA: $F = 0.41$, $df = 23$, $P = 0.80$). Estimates of ϕ also ranged widely, from 0.01 to 1.00 (data not shown), and modeling results from Program MARK indicated that stocking hatchery catchables also did not influence apparent survival; mean ϕ were 0.25 (SE = 0.22) and 0.27 (SE = 0.21) for control and treatment reaches, respectively (paired t -test: $t = 2.07$, $df = 23$, $P = 0.67$). Much of the difference between S (mean = 0.53) and ϕ (mean = 0.26) was probably attributable to annual PIT tag loss, which from 2006 to 2007 averaged 19% and ranged from 8% to 33%.

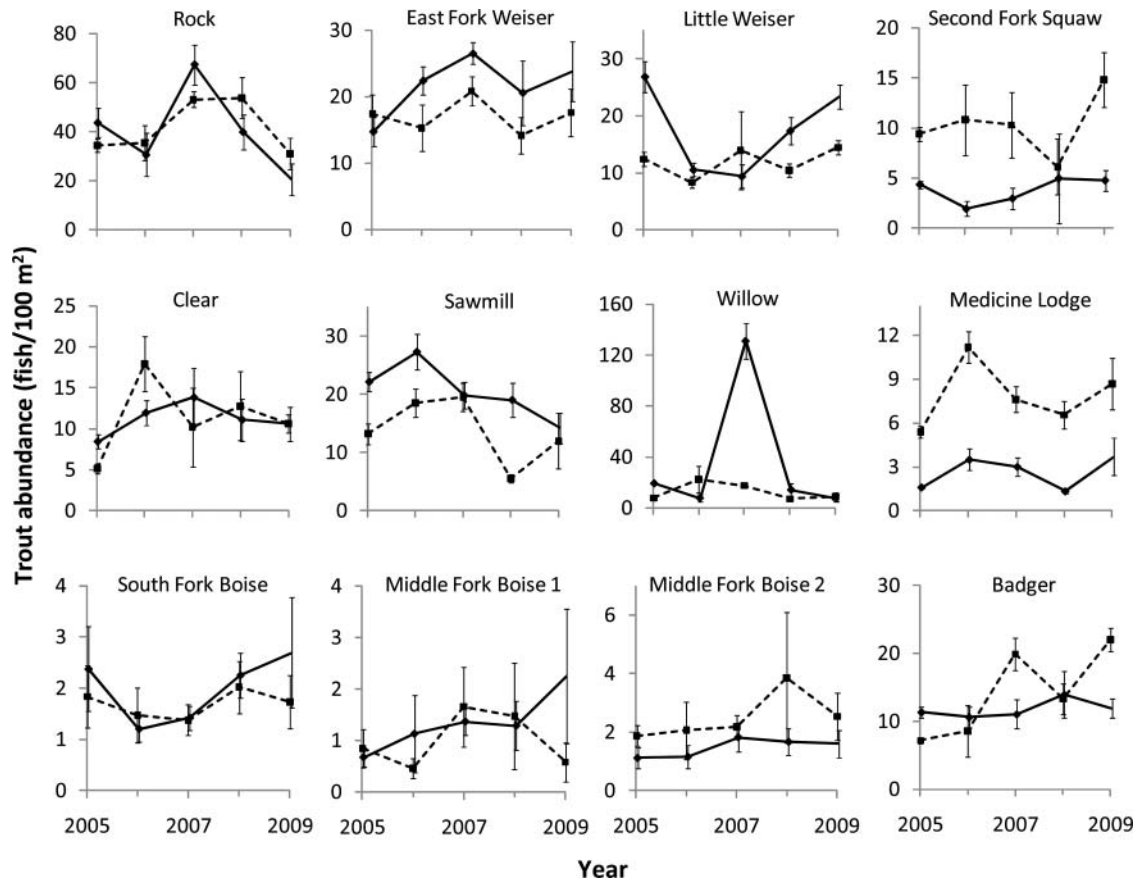


FIGURE 2. Estimated abundance of wild rainbow trout (fish/100 m²) in unstocked (solid lines) and stocked (dashed lines) reaches of streams from 2005 to 2009 in southern Idaho. Error bars represent 95% confidence intervals. Catchable stocking occurred from 2006 to 2008.

A total of 1,612 PIT-tagged wild rainbow trout were recaptured 1 year after initial tagging. On average, fish grew 44 mm from a given year to the next, and this growth appeared to be linear (Figure 6). Although growth was different from stream to stream (ANCOVA: $F = 972.8$, $P < 0.001$), there was no difference in growth between control and treatment reaches (ANCOVA: $F = 0.13$, $P = 0.72$). Based on otolith aging of PIT-tagged and other fish retained for age estimation in 2009 ($n = 471$), mean length (and range) at age 2 and age 3 at control reaches averaged 193 mm (146–264 mm) and 215 mm (163–324 mm), respectively, compared with 185 mm (150–245 mm) and 213 mm (167–264 mm) at treatment reaches. Mean length at age did not differ significantly between control and treatment reaches (paired t -test: $t = 2.12$, $df = 16$, $P = 0.58$). Relative weight also was not significantly different between control (mean $W_r = 86.3$; range, 79.4–96.7) and treatment (mean $W_r = 85.8$; range, 78.4–98.0) reaches (paired t -test: $t = 2.20$, $df = 11$, $P = 0.45$).

DISCUSSION

Our study provides tangible evidence that the stocking of catchable rainbow trout in southern Idaho streams had no affect

on the abundance, recruitment, survival, or growth of existing wild rainbow trout populations. Indeed, despite artificially increasing the abundance of rainbow trout in the treatment reaches by an average of 78% (range, 13–444%) via stocking, wild rainbow trout showed no ill effects at the population level. The lack of effect we observed on wild trout after stocking is most probably due to the poor competitive abilities of hatchery fish, and numerous studies support this conclusion. R. B. Miller was a pioneer in this work, and was involved in a series of investigations on wild and hatchery trout interactions (see Miller 1951, 1953, 1958). He reported that hatchery fish moved downstream after stocking but did not survive for long, especially overwinter. Hatchery catchables reared in streams fared better than pond-reared fish but not as well as wild fish transplanted from another location, and even the newly transplanted wild fish were out-competed by trout that already occupied the study area. Lactic acid levels were higher in hatchery fish than in wild fish and Miller concluded that hatchery fish died of exhaustion, probably not only from harassment but also from naivety in holding favorable stream feeding positions.

Since Miller's pioneering work, numerous additional studies have compared aggression, foraging behavior, movement,

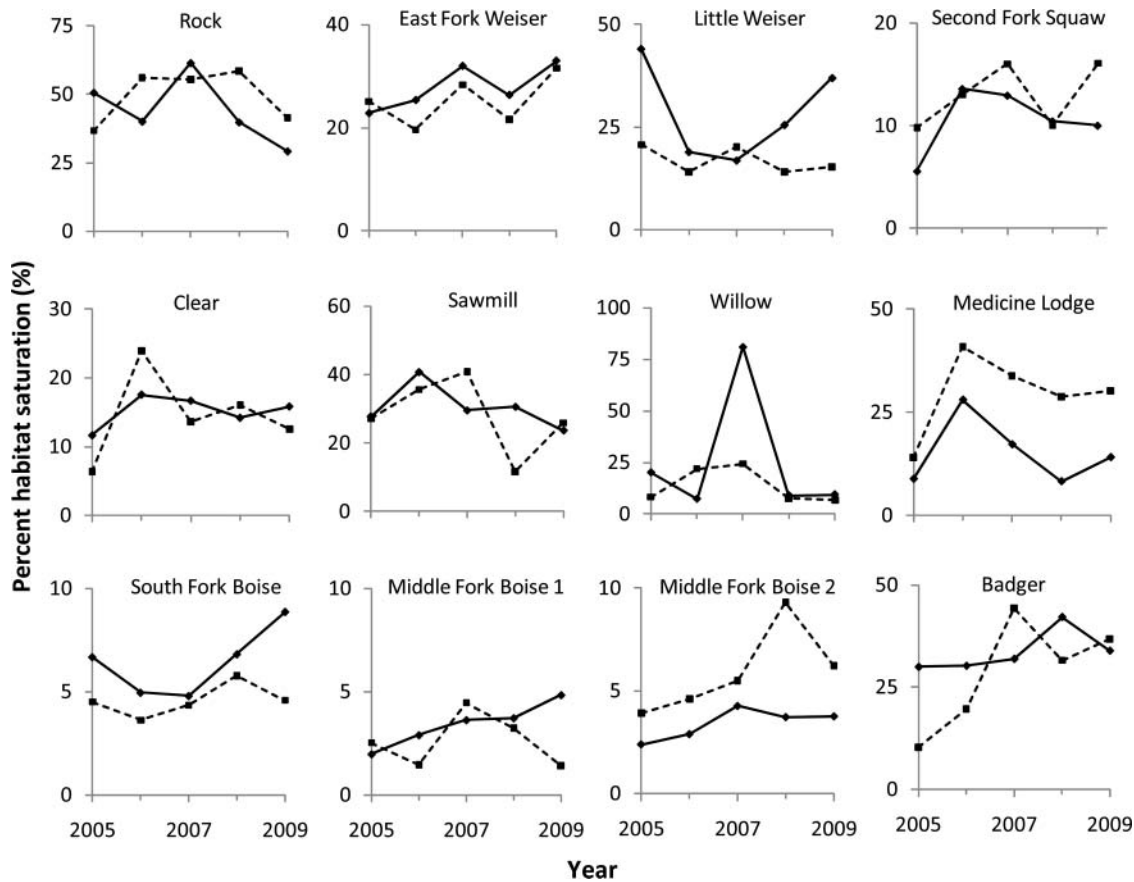


FIGURE 3. Estimated percent habitat saturation of wild rainbow trout in unstocked (solid lines) and stocked (dashed lines) reaches of streams from 2005 to 2009 in southern Idaho. Catchable stocking occurred from 2006 to 2008.

holding position, growth, and survival of hatchery and wild fish in an effort to infer competitive interactions (e.g., Needham and Slater 1945; Moyle 1969; Petrosky and Bjornn 1988; Mesa 1991; Peery and Bjornn 1996; Berejikian et al. 1999; see review in Weber and Fausch 2003). These and other studies have generally found that hatchery fish are more aggressive, use less energetically profitable holding and feeding positions, consume less food, and are less wary of predators, all of which appear to put them at a competitive disadvantage relative to wild trout. Vehanen et al. (2009) went so far as to suggest that, rather than hatchery fish causing distress in wild fish, the presence of wild trout may actually benefit hatchery fish because they learn feeding and holding behavior from wild fish; both Vehanen et al. (2009) and Huusko and Vehanen (2011) have provided some empirical evidence to support this.

It is unlikely that emigration of hatchery fish out of the stocked reaches would explain why we saw no population-level effects on wild rainbow trout populations. In a companion study, survival of stocked catchables in one of our study streams (Middle Fork Boise River) was short-lived (mean = 14 d) and dispersal 30 d poststocking was minimal (median = 100 m downstream; High and Meyer 2009). This lack of dispersal con-

curs with most previous work, which has shown that, in general, catchables disperse no more than about 1 km (Trembley 1945; Helfrich and Kendall 1982; Heimer et al. 1985; but see Bettinger and Bettoli 2002).

It is also unlikely that the lack of population-level effects on wild trout populations in our study was due to the use of triploid catchables rather than diploid fish. Previous research comparing the performance of triploid hatchery salmonids relative to diploids in natural environments has produced equivocal results (see Koenig et al. 2011), but most of the field evaluations have been conducted in lentic environments, where hatchery fish often persist for years (Teuscher et al. 2003; Koenig et al. 2011). In the only stream study we are aware of, diploid and triploid catchable-sized hatchery rainbow trout returned equally well to the creel of anglers, and the time of returns after stocking were equivalent (Dillon et al. 2000). Based on these similarities, and the abrupt poststocking mortality of catchables in streams both for diploids (Miller 1951, 1953; Walters et al. 1997; Bettinger and Bettoli 2002) and triploids (High and Meyer 2009), it is likely that diploid and triploid catchables would also have similar competitive abilities in stream environments, although this has never been tested.

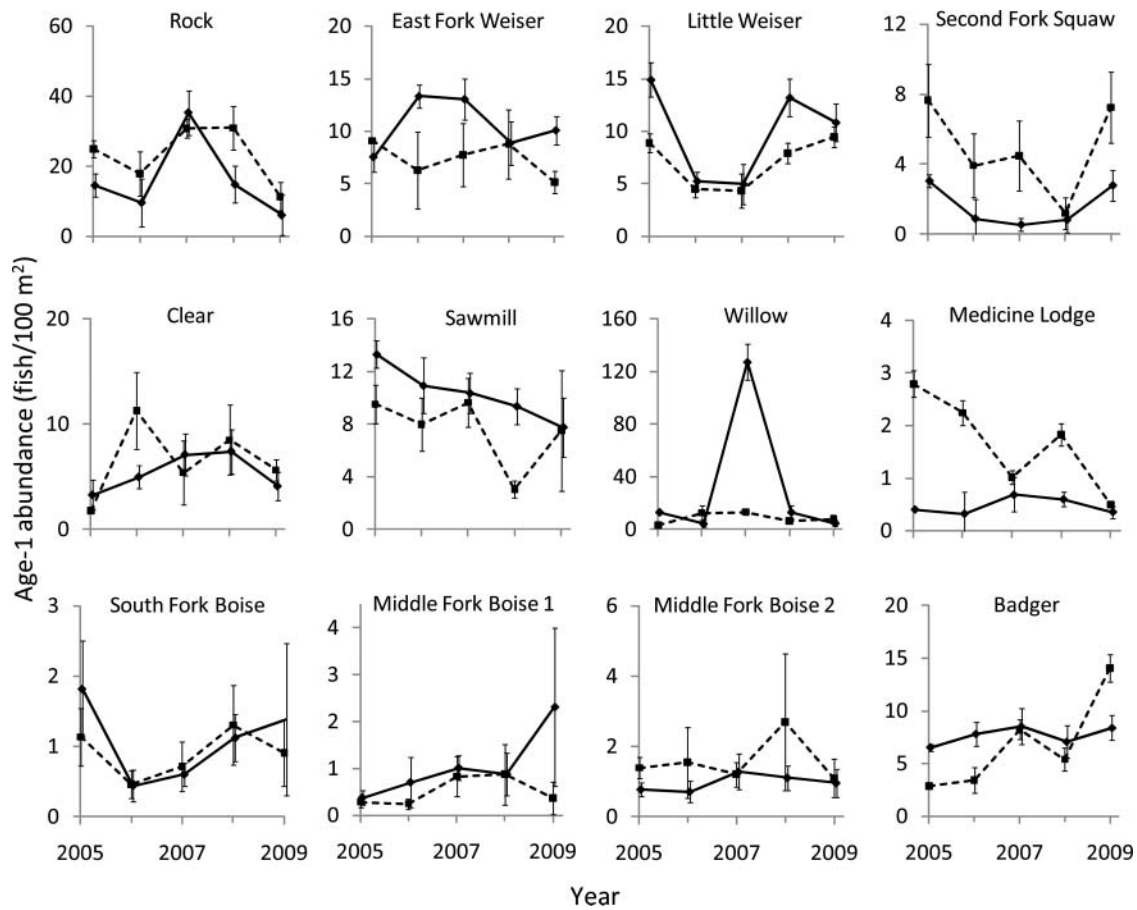


FIGURE 4. Estimated recruitment of wild rainbow trout (age-1 fish/100 m²) in unstocked (solid lines) and stocked (dashed lines) reaches of streams from 2005 to 2009 in southern Idaho. Error bars represent 95% confidence intervals. Catchable stocking occurred from 2006 to 2008.

Rarely has any study shown negative effects on wild trout at the population level owing to catchable stocking. One of the most cited examples is Vincent (1987), who reported that numbers and biomass of large, wild rainbow trout and brown trout (250–425 mm in length) decreased substantially in years when catchable rainbow trout were stocked in the Madison River and O'Dell Creek in western Montana, but smaller fish were not affected by stocking. However, a complicating factor in this study was that uncontrolled variables might have confounded the results. For example, winter flows were higher in the Madison River in the nonstocking years, potentially increasing wild trout abundance in the nonstocking years by increasing overwinter survival. In addition, exploitation was 50% greater during stocking years (presumably because anglers were attracted by the fish stocking), especially for larger-sized fish (Vincent 1980), which suggests that increased harvest also explained some of the reduced abundance of the larger (but not smaller) wild trout during stocked years. Other studies have also reported higher fishing mortality of wild trout populations associated with hatchery stocking (Butler and Borgeson 1965; Carline et al. 1991; Baer et al. 2007), and we speculate that increased angler effort and

harvest is usually the cause of decline in wild trout abundance associated with hatchery stocking, when such a decline occurs. This may explain why we saw no change in any of the vital statistics of wild trout in our study, since we did not advertise our stocking to anglers and, thus, they were unaware of the increased abundance of fish.

In our study, both estimates of survival were highly variable and thus within any given year were probably not very reliable, although the means ($\phi = 0.26$, $S = 0.55$) were within the range of typical stream-dwelling rainbow trout populations in Idaho (e.g., Schill 2009) and elsewhere (e.g., Cooper 2003). That ϕ was consistently lower than S was expected because a lost PIT tag was equated as a death in the CJS model, and annual tag loss in the first year of our study averaged 19% (Meyer et al. 2011). This finding emphasizes the often-overlooked necessity of estimating PIT tag loss during studies that use this tag in estimating population parameters for fish populations (see review in Dieterman and Hoxmeier 2009). In addition, for estimates of S (based on population estimates), wild trout emigrants may have balanced out immigrants, whereas for estimates of ϕ (based on PIT tag recaptures), emigrants could not balance out

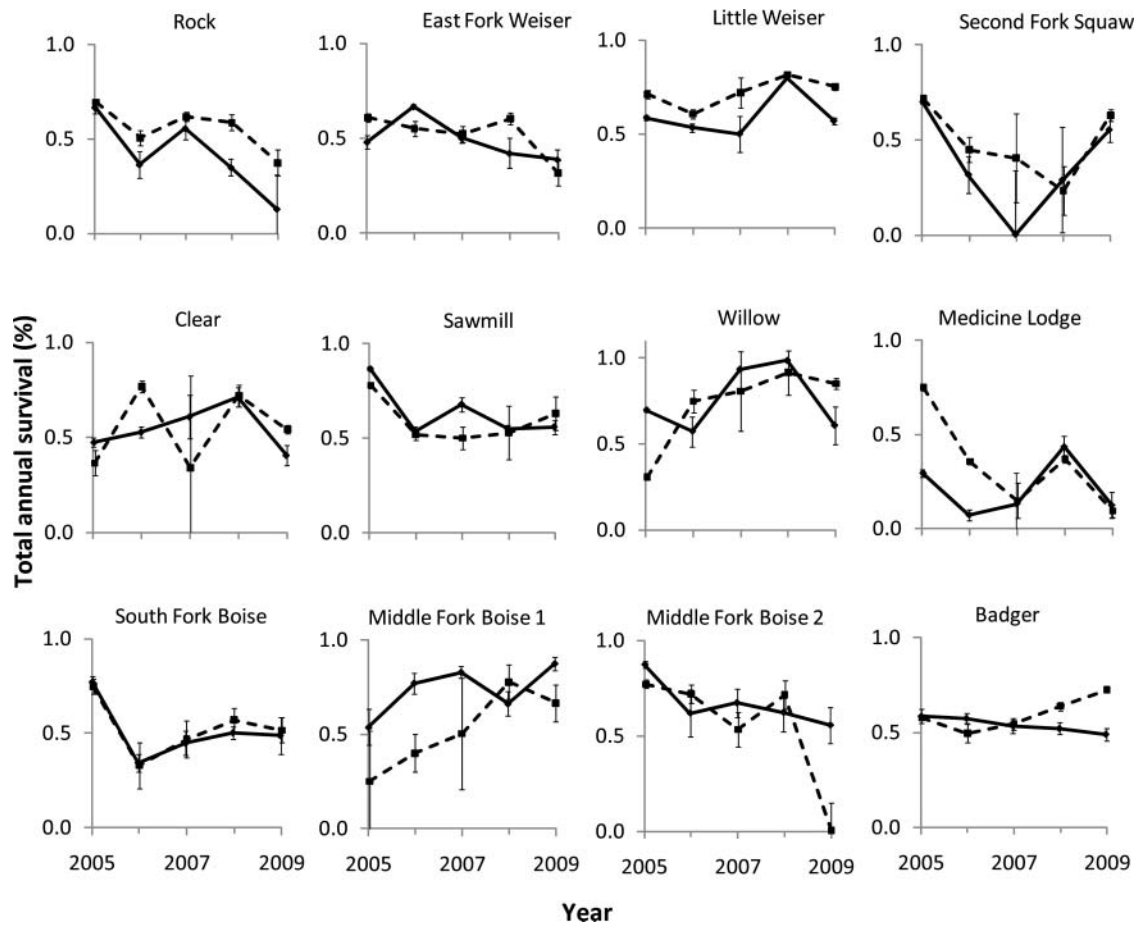


FIGURE 5. Total annual survival (S) of age-1 and older wild rainbow trout in unstocked (solid lines) and stocked (dashed lines) reaches of streams from 2005 to 2009 in southern Idaho. Error bars represent 95% confidence intervals. Catchable stocking occurred from 2006 to 2008.

immigrants since emigrants were not PIT tagged. Thus, any remaining difference between S and ϕ not attributable to PIT tag loss may represent the level of immigration and emigration that occurs among wild trout populations in southern Idaho. Regardless, the lack of effect on S and ϕ for wild trout suggests that not only was survival of wild trout unaffected by catchables, but stocking also did not cause any additional emigration when hatchery trout were stocked. Although displacement of wild fish by hatchery fish has been demonstrated in both laboratory and stream settings (see review in Weber and Fausch 2003), including at the reach scale (Symons 1969), our study reaches in general were nowhere near the spatial carrying capacity for wild trout (mean PHS = 20%). Thus, it is not surprising that the addition of hatchery fish caused no additional emigration of wild trout over a level that is considered normal for stream-dwelling salmonids.

We could not measure recruitment to age 0 because sampling in some streams occurred too early in the year to effectively capture these small fish. Instead, we inferred that any effect of stocking on wild rainbow trout spawning success or age-0 survival would have translated to lower abundance of

wild age-1 fish the following year. We could have missed an effect if recruitment was reduced by stocking, but this effect was obscured before we sampled age-1 fish the following year; density-independent survival constraints for age-0 fish during winter is one possible example. However, we argue that if such a “recruitment” effect was masked in all of our treatment reaches by the time fish reached age 1, then the effect to the population, though real, would still probably be meaningless in most instances.

Certainly, there are circumstances where stocking catchable trout can be detrimental to wild trout populations. First, and perhaps most obvious, is potential genetic effects, in which stocking hatchery fish can result in hybridization of native trout populations (Allendorf 1991; Allendorf et al. 2001). As mentioned above, hybridization with hatchery rainbow trout has already negatively affected several species of native salmonids in Idaho (Meyer et al. 2006; Kozfkay et al. 2011). However, since nearly all catchables stocked in Idaho are now treated to induce sterility, and induction rates are generally 100% both from IDFG egg production facilities (Kozfkay et al. 2006; T. Frew, IDFG, unpublished data) and from private facilities

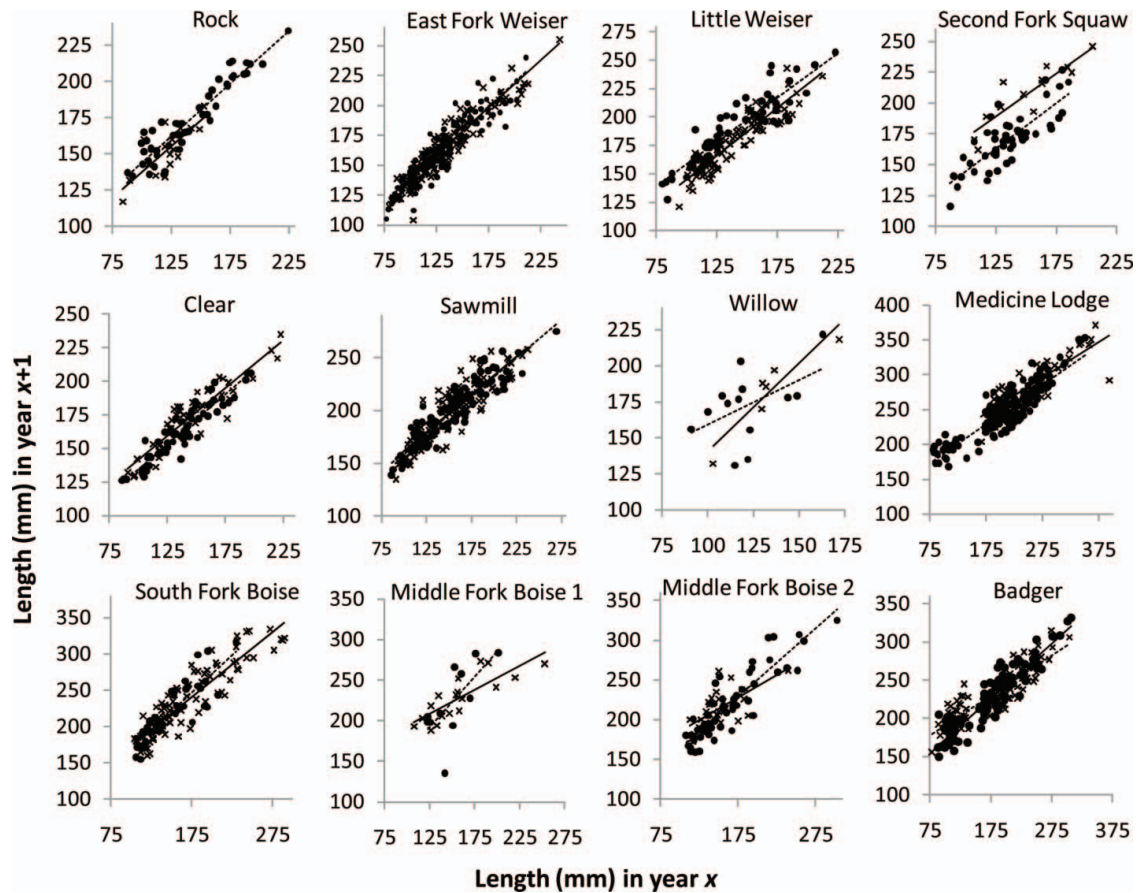


FIGURE 6. Growth (mm) of wild rainbow trout from year x to year $x + 1$ for PIT-tagged fish in unstocked (solid lines, dot marks) and stocked (dashed lines, x marks) reaches of streams in southern Idaho. Lines were fitted with least squares regression.

where all-female sterile eggs are purchased, the concern that continued stocking of catchables will result in further spread of hybridization has essentially been nullified in Idaho.

Second, although we saw no negative effect of stocking catchables on the vital statistics of wild trout populations, if we had stocked fish at higher densities, it is more likely that the catchables might have affected wild trout populations. We stocked streams at an annual stocking density of 4.2 fish/100 m², which is at the upper end of densities used by IDFG hatchery staff. We also stocked additional fish at the upper and lower reach boundaries, some of which probably drifted almost immediately into the treatment reaches (especially at the upstream boundary; High and Meyer 2009). By counting only the fish stocked in the middle of the treatment reaches, our stocking density was at the lower end of other comparable studies that have investigated wild versus hatchery competition in natural settings, such as those done by Carline et al. (1991; 8.1 fish/100 m²) and Vincent (1987; 16.3 fish/100 m²). Petrosky and Bjornn (1988) saw little effect of stocking catchables on wild rainbow trout until they used their highest stocking density, which, on average, was almost two orders of magnitude higher (253 fish/100 m²) than our stocking rate. Although the

above-mentioned negative effect of stocking catchables on wild trout shown by Vincent (1987) in Montana may instead have been largely due to the effects of stream flow and angler harvest, stocking rates were also almost four times higher than in our study. Including all catchables that we annually stocked, mean PHS in the treatment reaches was 48% (range, 26–94%), which, according to the logistic model of Grant and Kramer (1990), corresponded to a probability of observing a density-dependent response (in either growth, survival, or emigration) of 0.73 (range, 0.48–0.90). Although inclusion of all catchables certainly over-represents true PHS in treatment reaches because most catchables probably did not survive from one stocking event in any given month to the next (High and Meyer 2009), densities of fish were certainly high enough to expect some density-dependent response to stocking if competition for territorial space was occurring. Evidently our study reaches were not limited by rearing or holding habitat, but more likely by food, overwinter survival, spawning substrate, or some combination of these and other factors. Considering that hatchery trout are not efficient at foraging (Petrosky and Bjornn 1988; Mesa 1991) and would generally not be expected to survive long enough to compete for holding habitat in the coming

winter or for spawning habitat in the following spring (Petrosky and Bjornn 1988; Dillon et al. 2000), it is not surprising that no stocking effect was detected in the wild trout populations.

Besides stocking at higher densities, we could also have stocked for a longer period of time, and there is evidence that prolonged stocking may be more likely to negatively affect wild fish owing to cumulative effects. For example, Pearsons and Temple (2007) found little change in rainbow trout population abundance, size, or distribution after 5 years of supplemental stocking of juvenile Chinook salmon *O. tshawytscha*, but after 9 years, statistically significant decreases in abundance and biomass were more common (Pearsons and Temple 2010). However, as Pearsons and Temple (2010) noted, it seems likely that changes in abundance and growth would be more probable among small fish, which should have become manifest within a year or two in measurements of recruitment, and we detected no such effect. Moreover, Pearsons and Temple (2010) admitted that much of the increased detection of changes in fish populations with additional study years in their research might have been related to increased statistical power rather than cumulative stocking effects. We believe our study design had ample statistical power, with 12 paired control-versus-treatment comparisons over 5 years of monitoring, compared with the “increased statistical power” of Pearsons and Temple (2010) who had only two treatments and one control that were monitored for 9 years.

The apparently poor survival of catchables stocked in streams raises the question of whether stocking catchables is worth the expenditure of time and money to rear and stock them. In Idaho, streams stocked with catchables are expected to return 40% of the stocked fish to angler catch to justify the expenditure of raising and stocking them, but this is not always achieved. Dillon et al. (2000) estimated that 17% of stream-stocked catchable rainbow trout tagged with jaw tags (no rewards) were caught and reported by Idaho anglers, and considering that about one-half of anglers in Idaho report nonreward tags from the fish they catch (Meyer et al. 2010), this translates to an average harvest of about 34%. Most of the harvest observed by Dillon et al. (2000) occurred within the first 2–3 weeks after stocking, which is one reason why stream stocking in Idaho is usually spread out over three events during the angling season. High and Meyer (2009) suggested that stocking stream catchables should occur within 3 weeks of expected needs and within 1 km of the areas mostly heavily used by anglers to maximize return-to-creel rates in flowing waters.

In summary, despite the long-held notion that stocking catchable-sized hatchery trout negatively affects wild trout via competition (e.g., Butler 1974; Vincent 1974, 1987; Bachman 1984), relatively few studies have investigated competition effects at the population scale. Our results suggest that stocking hatchery catchables at densities at the upper end of existing IDFG stocking practices had no measurable effect on wild rainbow trout abundance, survival, growth, or recruitment in southern Idaho streams. Certainly, resource use overlaps to some degree between wild and hatchery trout in stream environments,

and our results do not lead us to advocate stocking hatchery fish haphazardly in streams across the landscape. Indeed, the century-long practice of stocking fertile nonnative trout in flowing waters throughout much of North America has been one of the primary causes of decline for many native salmonid species (Krueger and May 1991). However, when the risk of hybridization between stocked fish and native fish can be eliminated, we believe that stocking catchables at normal stocking densities will have negligible effects on existing populations of wild stream-dwelling trout.

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